Reducing Variability in Freshwater Macroinvertebrate Data

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Abstract
The benthic macroinvertebrate community is often used to evaluate stream water quality, but this efficiency of this process may be complicated by high data variability. This variability can be reduced by proper selection of sampling sites, collection methods, identification levels, and analysis metrics. Corrections also can be made to compensate for predictable changes associated with ecoregion, stream size and seasonality. Some evaluation should be made for the effects of antecedent flow, especially after droughts and high rainfall periods.

Key words: North Carolina, benthos, data variability, methods, identification.

Introduction

Environmental monitoring groups often use characteristics of freshwater macroinvertebrate communities to assess stream water quality. In cases of severe pollution, any kind of collection technique and/or any kind of data analysis can be used to demonstrate a water quality problem. In cases of "less than catastrophic" pollution, however, high data variability may obscure the effects of changes in water quality (Howmiller 1975). There are many different sources of variation for benthic macroinvertebrate data, including differences in collection efficiency, habitat, season of the year, and flow.

The problems of data variability can be greatly reduced by making corrections for any changes in habitat and season of the year, as well as through wise choices of identification levels, collection methods, and data analysis techniques. Erman (1981) has shown the frustrations in trying to compare studies with different collection techniques and identification levels. This paper will focus on North Carolina's experience with making these choices, and the ways we are developing seasonal and habitat-associated adjustments to our biocriteria. Some overlap with Lenat (1988) is inevitable, as both papers discuss the subject of taxa richness variability, but a large amount of new material has been included.

The North Carolina program was originally set up to deal with relatively simple between-station and between-date comparisons; the emphasis was on showing large changes in water quality or habitat quality. As the water quality program expanded, we began to look at more subtle water quality problems. Monitoring was required for all stream sizes (from temporary streams to large rivers) and we were asked to make collections during all months of the year and under a variety of flow conditions. To deal with these complicating factors, we are examining "normal" changes in the benthic macroinvertebrate community associated with differences in habitat, stream size, and seasonality.

North Carolina originally used quantitative collections (kick-net samples) to evaluate the benthic macroinvertebrate community. All samples were laboriously sorted in the lab. As our monitoring requirements expanded, we developed several new collection methods to collect reliable information in a more cost-efficient manner, including a new "rapid bioassessment" technique.

Much of the data presented in this paper is still in a preliminary stage of analysis, as North Carolina has just completed a four month effort to put all information (1983-present) into a large computerized data base. We have been using this data set to look at the spatial, temporal preferences of each taxa, as well as generating pollution tolerance data. We would like to use this paper as a means of soliciting opinions and advice concerning these analysis methods from other biomonitoring groups.

Results and Discussion

Collection and Identification Choices

The first step in reducing variability is to apply common-sense during sample collection. Stations should be chosen to be similar in habitat characteristics, and collections should not be made if high flow will interfere with collection efficiency. The collection method also should be suitable for the habitat being sampled. For example, dredge samples are rarely appropriate for shallow, fast-flowing streams.

There is considerable disagreement about the appropriate identification level and/or what groups should be identified (see Lenat 1988). North Carolina has chosen to use species or genus level identifications (where possible), including the infamous Chironomidae. It is clear that species level identifications increase the efficiency of site classifications (Resh and Unzicker 1975, Furse et al. 1981, Furse et al. 1984, Hilsenhoff 1982, Rosenberg et al. 1986), but with a cost of added identification time. I agree with Hilsenhoff (1982) that the added time required for species identifications is trivial compared to collection and sorting time. Many investigators elect to identify the Chironomidae to family (or subfamily) level, even if other groups are classified at a genus/species level. While the taxonomy of this group can be difficult, the information added by good chironomid data can be valuable in determining the nature of water quality problems.

Collection methods should be chosen which yield reliable data in the most cost-efficient manner. This choice will vary depending on the objectives of the study, especially on the need for precise estimates of species abundance. Abundance measurements will be required for life cycle studies and production studies, but are notoriously difficult to obtain. Our experience in water quality assessment is that we need a quantitative estimate of taxa richness and a qualitative estimate of abundance values (Rare, Common, Abundant). These requirements lead to the derivation of our standardized qualitative collection method (Lenat 1988).

All North Carolina collection methods utilize large composite, multiple-habitat, samples. The Standard qualitative method utilizes 10 samples, taken with 6 different collection methods. We have also developed an Abbreviated ("rapid bioassessment") collection method, which has become an important part of North Carolina's biomonitoring program. The latter method uses only 4 composite samples (kick-net, sweep-net, leaf-pack, and "visuals"), with collection and identification limited to the EPT groups. Note that the Abbreviated collection method produces a sub-sample of the Standard collection. We have recently compared Standard and Abbreviated samples collected independently at 30 sites (Larry Eaton, unpublished data). The 4-sample collections naturally collect fewer species than the 10-sample collections, but results from these two methods are highly correlated (Figure 1, r^2=0.96), allowing criteria to be developed for each. High variability is associated with a smaller sample size, but this is offset by the larger number of sites that may be sampled. A more detailed description of the Abbreviated method is in preparation.

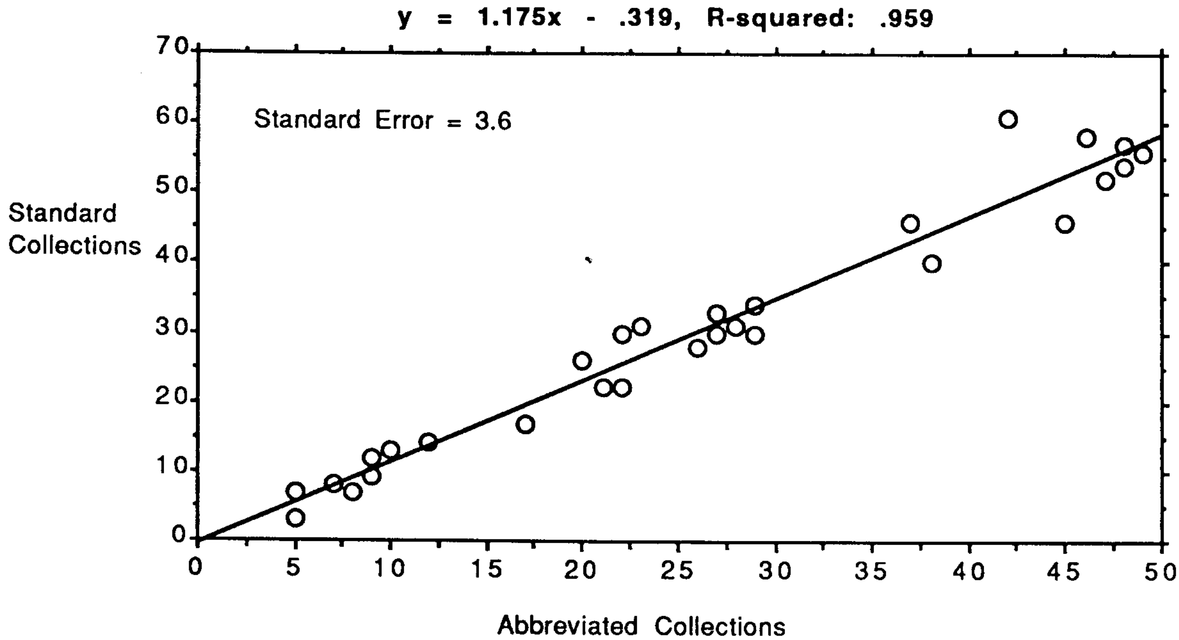

Figure 1. EPT taxa richness for standard (10-sample) collections vs. abbreviated (4-sample) collections.

Analysis Metrics
The choice of analysis metrics have a significant effect on the variability of your data or the reliability of site ratings. The ideal metric will be insensitive to normal habitat changes, but sensitive to changes in water quality. Many monitoring groups are trying to increase the confidence in their water quality evaluations by using several (relatively independent) ways of examining the benthic macroinvertebrate community. This latter technique has been borrowed from the Index of Biotic Integrity (Karr 1981) used by fisheries scientists.

Taxa Richness. The North Carolina methods tend to focus on taxa richness, especially taxa richness for the intolerant (EPT = Ephemeroptera + Plecoptera + Trichoptera) groups. Many investigators have shown that taxa richness (and related parameters) are more stable than abundance values (Godfrey 1978, Minshall 1981). Taxa richness values have been frequently associated with environmental stress (especially water quality), but this parameter is fairly stable in clean water habitats, even given some changes in habitat characteristics and/or flow (Patrick 1975, Bradt and Wieland 1981, Minshall 1981, Wagner 1984).

Biotic Indices. Another way to reduce variability is to use metrics which are (theoretically) independent of sample size. Diversity indices were derived with this in mind, but have proved to be unreliable in many types of pollution assessment (Godfrey 1978, Hughes 1978). Biotic indices have greater promise for water quality assessment (Hilsenhoff 1982), but their use in the Southeast has been hampered by the lack of a good data base on the environmental tolerances of benthic macroinvertebrates. Tolerance values have invariably been

Table 1. Preliminary information for deriving a North Carolina biotic index from existing bioclassifications. Mean abundance values vary from 0-10 and bioclassifications are coded 1-5. Percentile calculations are based on cumulative abundance values, starting from the Excellent bioclassification.

Bioclassification:	Poor	Fair	Good-Fair	Good	Excellent	Bioclass # Mean	Percentiles 75th	Percentiles 90th	Converted[1] 75th
	Mean Abundance Values								
Bioclass #:	1.0	2.0	3.0	4.0	5.0				
Percentile									
Intolerant Species									
Drunella wayah	-	-	-	0.2	0.3	4.5	4.6	4.3	0.6
Rhithrogenia spp.	-	-	0.1	0.2	0.8	4.5	5.0	4.0	0.0
Chimarra spp.	0.1	0.3	1.2	2.5	3.8	4.2	4.2	3.0	1.1
Micrasema wataga	-	0.1	0.8	0.8	1.0	3.9	3.7	3.2	1.9
Goera spp.	-	-	-	0.3	0.6	4.5	4.8	4.3	0.3
Brachycentrus chelatus	-	-	-	0.1	0.1	4.5	4.5	4.2	0.7
Pteronarcys dorsata	-	0.1	0.3	0.7	0.7	4.0	4.1	3.3	1.3
Acroneuria abnormis	0.2	0.6	2.3	5.4	8.0	4.2	4.2	3.0	1.1
					Means:	4.3	4.4	3.7	0.6
Facultative Species									
Stenonema modestum	1.5	7.0	8.4	7.8	8.3	3.5	3.0	2.2	2.9
Ephemerella catawba gr.	0.7	0.7	0.9	1.7	3.0	3.9	3.4	2.0	2.3
Eurylophella temporalis	0.1	0.4	0.9	1.3	1.3	3.8	3.5	2.7	2.1
Cheumatopsyche spp.	3.0	7.3	7.7	7.0	7.4	3.4	2.7	2.0	3.3
Hydropsyche venularis	0.7	1.9	3.4	2.5	2.7	3.5	3.1	1.9	3.0
Perlesta spp.	0.1	1.0	1.4	1.6	1.4	3.6	3.2	2.4	3.1
Ancyronyx variegata	0.9	2.1	2.2	1.5	0.9	3.1	2.5	1.8	3.6
Polypedilum convictum	0.5	1.6	2.8	2.0	1.8	3.4	3.0	2.0	2.9
					Means:	3.5	3.1	2.2	2.9
Tolerant Species									
Cricotopus bicinctus	3.6	3.3	3.2	2.0	1.1	2.8	1.9	1.4	4.4
C. tremulus gr (C/O sp. 5)	1.4	1.2	1.1	0.7	0.4	2.7	1.8	1.2	4.6
Chironomus spp.	3.8	2.2	1.5	0.9	0.5	2.4	1.6	1.2	4.9
Polypedilum illinoense	4.3	3.4	3.3	2.3	1.7	2.9	1.9	1.3	4.4
Physella spp.	3.4	2.7	2.3	1.8	1.1	2.8	1.8	1.3	4.6
Argia spp.	3.1	3.4	2.7	1.9	1.8	2.9	2.0	1.4	4.3
Limnodrilus hoffmeisteri	3.6	2.2	0.8	0.6	0.6	2.3	1.5	1.2	5.0
Asellus spp.	1.7	1.3	0.8	0.5	0.3	2.5	1.7	1.2	4.7
					Means:	2.6	1.8	1.3	4.6

[1]Numbers "flipped" so that a higher value reflects greater pollution tolerance: x = 6-y, range expanded (with regression equation) to a 0-5 scale: tolerance value = 1.43x - 1.43. Converted numbers are comparable to a Hilsenhoff-type index.

assigned based on best professional judgement, as was the case for North Carolina's existing biotic index.

North Carolina has initiated a program to more systematically derive invertebrate tolerance values, using our existing computerized data base. This data base currently has 1300+ individual collections, including samples from a broad range of water quality classifications, ecoregions, stream sizes and seasons. Table 1 presents some very preliminary data from our efforts to derive tolerance values. I present this information here in an effort to solicit comments and suggestions from readers, the final form of our biotic index may vary substantially from the concept presented here.

The initial step was to combine information on bioclassifications (based on EPT taxa richness), with abundance (0=Absent, 1=Rare, 3=Common, 10=Abundant) and frequency data. The first set of numbers in Table 1 are average abundance values (0-10) for each water quality class. The summary values are based on the water quality class (1-5), with a mean, 75th percentile and 90th percentile. Percentiles are based on the cumulative frequency distribution, starting from Excellent water quality (Class #5). Ideally, the tolerance values should show a large separation of tolerant and intolerant taxa, while still producing intermediate values for facultative taxa (near the median bioclass # of 3.0). The 75th percentile number was chosen as the summary statistic closest to these ideal characteristics, and was converted to a Hilsenhoff-type biotic index. The numbers were "flipped" so that a higher number reflects greater pollution tolerance. This produced a range of values similar to a Hilsenhoff index, but with a range of only 1.0 to 4.5. A simple regression equation was used to expand this range to 0-5, with the resulting numbers directly comparable to Hilsenhoff-type indices. If there is insufficient data to derive a tolerance value for some species, the original value (based on best professional judgement) can be retained. This approach to deriving a biotic index seems to show great promise as an alternate method of bioassessment. The next step would be to derive index criteria for both Standard and Abbreviated samples.

Collector Effects

Several investigators have examined the effect of the collector on the variability of benthic invertebrate data (Chutter and Noble 1966, Pollard 1981, Furse et al. 1981, Lenat 1988). While some differences can be found, most studies agree with Egglishaw (1964) that collector effects are "not large". The type of differences noted by Furse et al. (1981) argue strongly for standardization of collection methods.

Habitat Effects

Between-site and between-sample differences in habitat often contribute to data variability. If these differences are large, it may negate any attempt to look for changes in water quality. In many cases, the investigator can limit habitat differences, but these problems may be unavoidable for basin-wide surveys. Habitat differences can be considered for three distinct size scales: ecoregion, stream reach, and microhabitat.

Ecoregion. Ecoregion is rapidly becoming one of the most significant buzz-words of the 1990's. No government document can be released without at least one reference to the need for ecoregion reference sites. The ecoregion concept suggests that streams within a relatively uniform geographic areas will have similar faunas, or at least similar community structure (Hughes and Larsen 1988). This concept has been most fully developed for fish communities, but has also been shown to be applicable to stream invertebrates (State of Arkansas 1987, Lenat 1988). North Carolina has utilized three broad eco-

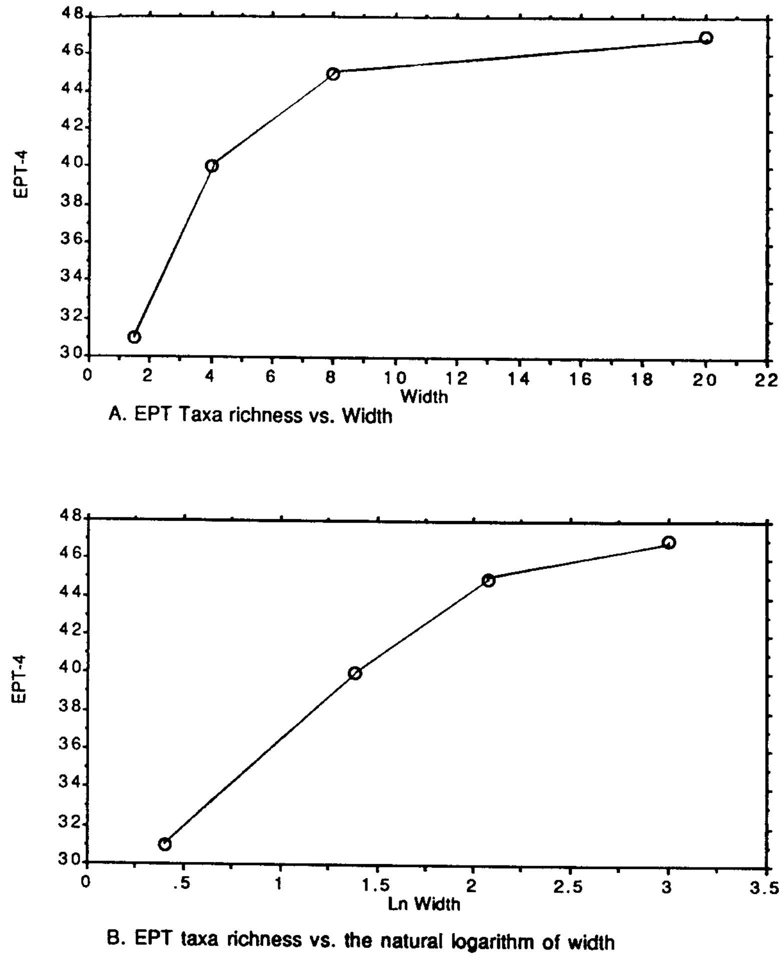

Figure 2. EPT taxa richness (abbreviated samples) vs stream width (m). Cataloochee Creek catchment, January 1990.

regions to develop bioclassification procedures, but there may be up to 12 different ecoregions in our state. Preliminary work indicates that at least 7 ecoregions will be needed to establish reliable site classifications, requiring up to 7 different sets of biocriteria. Some important factors in determining ecoregion include elevation/slope, soil type and permeability, geology, vegetation, and land use.

Stream Reach. At the next size scale, one must consider variability between stream reaches, especially in regard to stream size. Several studies have looked at the changes in the invertebrate community in relation to stream size, usually indicating an increase in taxa richness from first to fifth order streams, with a decline in higher order streams. Such studies usually look at average values per sample, rather than looking at changes in the entire stream community (Minshall et al. 1985 and Naiman et al. 1987). It is possible that a part of the decline in higher order streams is related to the smaller proportion of the stream that single (usually midstream) collections will sample in larger rivers. Gaschignard et al. (1983) found that the river fauna could be separated into two units: a mid-channel community, and a community found within 10 meters of the bank. In small streams, midchannel samples will include both assemblages. As stream size increases, however, there is a decreased probability that the bank assemblage will be included in single-habitat samples. Multiple-habitat samples may eventually produce a slightly different picture of stream size versus taxa richness.

All investigators agree that lower taxa richness is expected in small stream. This point is illustrated in Figure 2, showing a sharp drop in taxa richness in comparing a site 1.5 meters in width with a site 4.5 meters in width. Taxa richness vs. the natural logarithm of width (in this example) showed an almost linear relationship. Most biological criteria are derived from larger streams and rivers; a logical refinement would be to make some adjustment for different size classes.

The problem of classifying streams with taxa richness values is greatest for very small streams. These streams will have more limited habitat complexity, but the most important cause of reduced taxa richness in these systems is the periodic stress caused by drought/low flow conditions. Droughts may cause drastic reductions in current speed, often with an accompanying reduction in dissolved oxygen; some streams may dry up entirely.

What constitutes a "small stream" in North Carolina will vary with soil permeability. In well-drained soils (Sandhills ecoregion), permanent flow occurs in some streams less than one meter wide. In poorly drained soils, however, (Slate Belt Ecoregion) streams up to 15 meters wide may become temporary during extended droughts. In evaluating very small streams, it is important to evaluate prior flow/rainfall records.

Small pristine mountain streams also have been found to have reduced taxa richness and North Carolina is in the process of deriving special criteria for these areas. Preliminary analysis indicated that these criteria should be applied only to mountain streams with the following physical characteristics:

1. First or second order stream

2. Average width <4 meters

3. Largely closed canopy (70-100%)

4. No abundant Aufwuchs growths

Given these characteristics, we would define areas with an Excellent bioclassification based on EPT taxa richness (>27 for Abbreviated samples, >30 for Standard samples), ratio of EPT S/Total S (>0.5), Few Odonata, Coleoptera and Mollusca (<10% of total taxa richness), a biotic index value (still being derived) and the presence of species characteristic of small streams. A list of "small stream" taxa also is currently being developed from our data base. All of the above classification criteria are in review, and some minor changes are expected.

Microhabitat. Examination of individual samples has often indicated species with a "clumped" spatial distribution. This problem can be overcome by the use of larger samples, especially composite samples. This is the strategy implicit in "traveling kicks", many types of D-frame or pond-net collections, and North Carolina's composite collections. Our multiple-habitat semi-quantitative sampling should help to reduce microhabitat variations.

Jenkins et al. (1984) recommended sampling at least three habitats to adequately inventory the aquatic fauna, especially in relation to the "conservation" value of streams. Brooker (1984) also showed that the effects of habitat change (channelization, etc.) were not properly assessed by riffle-only collections. Cuff and Coleman (1979) showed that overall precision was increased by taking single samples from many stations, rather than by taking many replicates at a single site. This analysis would seem to support a multi-habitat sampling design.

Changes with Time

Seasonal Changes. Individual macrobenthic species are well known to exhibit marked seasonal changes in abundance (Hynes 1972). Overall seasonal changes in community structure are more difficult to form generalizations about, but we should expect considerable between-ecoregion and between-year differences, largely due to differences in seasonal temperature regimes. Spring and/or fall peaks in taxa richness have been observed at many of our North Carolina sites, with the spring peaks being the most pronounced. Seasonality changes are not predictable using a "standard" correction factor for each month. Different years may have quite different seasonal patterns, especially with regard to the onset of spring generations. We have also found that greatest seasonal variation occurs at sites with highest water quality, i.e., seasonal variation is reduced at severely polluted sites. Some of the "seasonal" change in slightly impacted streams may reflect a real change in water quality, not a change caused by temperature-related hatching or emergence. The latter is especially true in agricultural areas, where there may be a seasonal input of sediment, nutrients and/or pesticides.

The first step in making seasonal corrections in taxa richness is some knowledge of the life cycles of the invertebrates in each ecoregion (Table 2). Year-round species, or multi-voltine species with no resting stage, have little influence on seasonal changes in taxa richness. However, many species will be absent for a portion of the year, sometimes up to 9 months. Often spring peaks in EPT taxa richness are caused by the addition of many Plecoptera species. This pattern is illustrated in Table 3, comparing EPT taxa richness of single spring collections with average summer data. It is apparent from these examples that a large part of the spring taxa richness increase was caused by the appearance of many plecopteran taxa. In some cases, some adjustment also must be made for increases in Ephemeroptera. Simple subtraction of these species, rather than making the same proportional adjustment for all sites, appears to be the most reasonable means of seasonal adjustment. In all cases, the seasonal adjustment must be validated by comparison with summer data. We have not yet been able to come up with an adjustment scheme that does not require such test sites. The importance of control sites, especially ecoregion reference sites, cannot be overemphasized in making water quality assessments outside of the usual summer collection periods.

Flow. Some "seasonal" changes do not reflect normal shifts in populations, but irregular changes in water quality or habitat quality, often related to flow. Given adequate flow information, it may be possible to predict at least the direction of changes associated with floods and/or droughts. Note that high quality (daily/hourly) flow information is usually available from the United States Geological Survey's monitoring network.

Extreme variation in flow has been shown to have a catastrophic effect on the macroinvertebrate fauna of some streams (Gray 1981). Given some refuge from scouring, however, the invertebrate community can withstand more moderate changes in flow. Data from both King (1983) and Poole and Stewart (1976) indicate that the hyporheic zone may act as a partial refuge from the effects of elevated flow. The invertebrate community, however, seems to have much of its variability caused by changes in flow (Leland et al. 1986, McElravy et al. 1989); some seasonal minima may be more related to floods than to emergence (Chutter 1970).

The effects of drought and flood are often very site-specific, but can be

Table 2. Examples of variations in normal seasonal patterns.[1] Numbers are frequency of collection (0-1) x average abundance value when present (0-10), final values vary from 0-10. Underlining indicates periods of maximum abundance, bold-faced type used to show minima.

A. Year-round taxa: Multiple species/Univoltine or multivoltine with no resting stage

	1	2	3	4	5	6	7	8	9	10	11	12
Stenonema modestum	6.5	6.6	6.3	**5.0**	6.8	6.7	8.1	8.2	6.7	8.2	**5.0**	7.6
Acroneuria abnormis	3.2	2.4	3.3	2.7	**2.1**	**2.0**	3.5	4.8	3.6	4.5	**2.4**	**2.4**
Stenacron interpunctatum	**0.4**	**0.8**	1.5	2.0	3.7	1.2	3.0	3.4	2.0	2.0	1.4	1.3
Isonychia spp.	3.4	1.9	2.7	2.0	3.0	4.3	5.3	6.4	3.3	4.1	2.4	2.3
Hydropsyche sparna	1.9	1.3	3.1	1.2	1.4	1.6	2.3	3.7	1.1	2.0	1.3	1.3
Cheumatopsyche spp.	4.7	4.0	6.0	3.7	7.4	5.7	7.4	8.0	5.5	6.3	4.5	9.8

B. Almost Year-round species, with periods of distinct absence or minima.

	1	2	3	4	5	6	7	8	9	10	11	12
Baetisca carolina	1.1	0.4	0.3	0.6	0.1	0.4	+	+	0.3	0.2	0.7	0.8
Caenis spp.	+	0.7	0.7	1.6	1.9	2.9	3.5	2.2	1.8	0.6	1.1	**0.2**
Serratella deficiens	0.3	0.7	0.3	1.2	2.5	0.9	1.6	2.3	**0.1**	**0.3**	**0.2**	0.4
Heptagenia marginalis	**0.2**	**0.1**	**0.3**	+	0.4	0.5	1.6	2.5	1.2	0.6	0.3	+
Eurylophella temporalis	1.2	1.8	3.4	1.5	3.2	1.0	**0.1**	**0.1**	**0.2**	0.5	0.6	1.5
Neoperla spp.	0.6	0.4	+	+	0.8	0.3	0.5	0.1	0.5	1.8	0.3	0.3
Perlesta spp.	**0.2**	0.8	1.0	3.0	5.4	3.1	1.1	0.4	+	**0.1**	**0.2**	0.6
Trianenodes tarda	**0.2**	**0.2**	+	0.7	0.4	1.0	1.4	1.0	0.6	1.2	**0.1**	0.7
Hydroptila spp.	**0.2**	**0.2**	**0.3**	0.6	0.3	0.9	1.1	1.5	0.3	**0.2**	**0.3**	0.6
Hydropsyche morosa	**0.2**	+	**0.2**	0.4	-	0.2	1.0	1.9	**0.2**	**0.3**	+	0.4

Fast (Short Life Cycle) Taxa: Univoltine with resting stage.

	1	2	3	4	5	6	7	8	9	10	11	12
Danella simplex	-	-	-	-	-	+	0.3	0.9	-	-	-	-
Drunella allegheniensis	-	-	-	-	-	+	0.3	0.9	-	-	-	-
Serratella serrata	-	-	-	-	0.2	0.2	0.1	0.4	+	-	-	-
Baetis pluto	0.2	+	+	0.8	1.8	1.5	1.0	2.0	1.8	3.5	0.7	0.1
Cinygmula subaequalis	-	-	0.7	1.9	0.1	0.3	-	-	-	-	-	-
Drunella walkeri	-	-	0.9	1.0	0.3	0.5	-	+	-	-	-	-
Agapetus spp.	0.4	0.1	0.3	0.4	0.6	0.4	+	+	-	-	-	-
Isoperla namata	1.0	1.0	3.9	1.2	-	0.1	-	-	-	-	-	0.5
Clioperla clio	1.2	1.7	0.3	-	+	-	-	-	-	0.6	1.2	1.5
Leptophlebia spp.	2.3	2.4	1.0	0.1	0.1	+	+	0.1	+	1.1	2.4	3.7
Apatania spp.	4.0	0.7	0.2	-	-	-	-	+	0.2	0.3	1.5	2.0
Strophopteryx spp.	5.1	3.7	0.9	+	-	-	-	-	-	-	0.9	2.6

[1]Numbers are derived from North Carolina's computer data base (1983-present, 1300+ collections), representing a wide range of water quality conditions, ecoregions, seasons, and stream sizes.

Table 3. Evaluation of EPT taxa richness, comparing summer vs.spring collections in three ecoregions of North Carolina.

	Summer Value Mean (Range)		Spring Value		# Univoltine Taxa with (<6 month) Life Cycles Summer	Spring
A. Mountain						
French Broad River at Rosman						
Ephemeroptera	20.3	(19-23)	22	(No change)	8	8
Plecoptera	7.0	(6-8)	14	(+7)	0	11
Trichoptera	17.0	(12-20)	19	(No change)	4	3
Total	44.3		55	(+11)		
B. Upper Piedmont						
Mayo River at Price						
Ephemeroptera	18.0	(-)	23	(+5)	9	11
Plecoptera	4.5	(3-6)	13	(+8)	0	8
Trichoptera	16.0	(-)	18	(No change?)	6	3
Total	38.5		54	(+15)		
C. Coastal Plain						
Drowning Creek near Hoffman						
Ephemeroptera	6.5	(5-8)	11	(+4)	2	3
Plecoptera	6.5	(6-7)	12	(+5)	0	7
Trichoptera	16.5	(15-19)	17	(No change)	1	2
Total	29.5		40	(+10)		

broken down into a series of common sense questions:

1. Was there a substantial decline in current velocity that might eliminate high current species? (especially in small streams)

2. Was there a change in scour? (especially for extremely sandy streams with little or no refuge) Was there a refuge from scour and was this refuge included in the samples collected? Refuges include interstitial habitat (especially clean rubble/boulder substrate), snags above the bottom, river weed, etc.

3. Was there a change in dilution of a point source discharger, especially if organic loading was a problem? If there was a significant point source impact, was there a change in length of recovery zone? Note that recovery zones are often shorter under low flow conditions, but with more acute effects close to discharge point.

4. Was there a change in the amount of nonpoint runoff, especially if the catchment contains land-disturbing activities?

5. Was there a change in macrophyte growths or the Aufwuchs population caused by a change in transparency, scour, and/or nutrient concentration?

Separating out the possible effects of changes in flow regimes from real changes in water quality is the task of most trend monitoring networks.

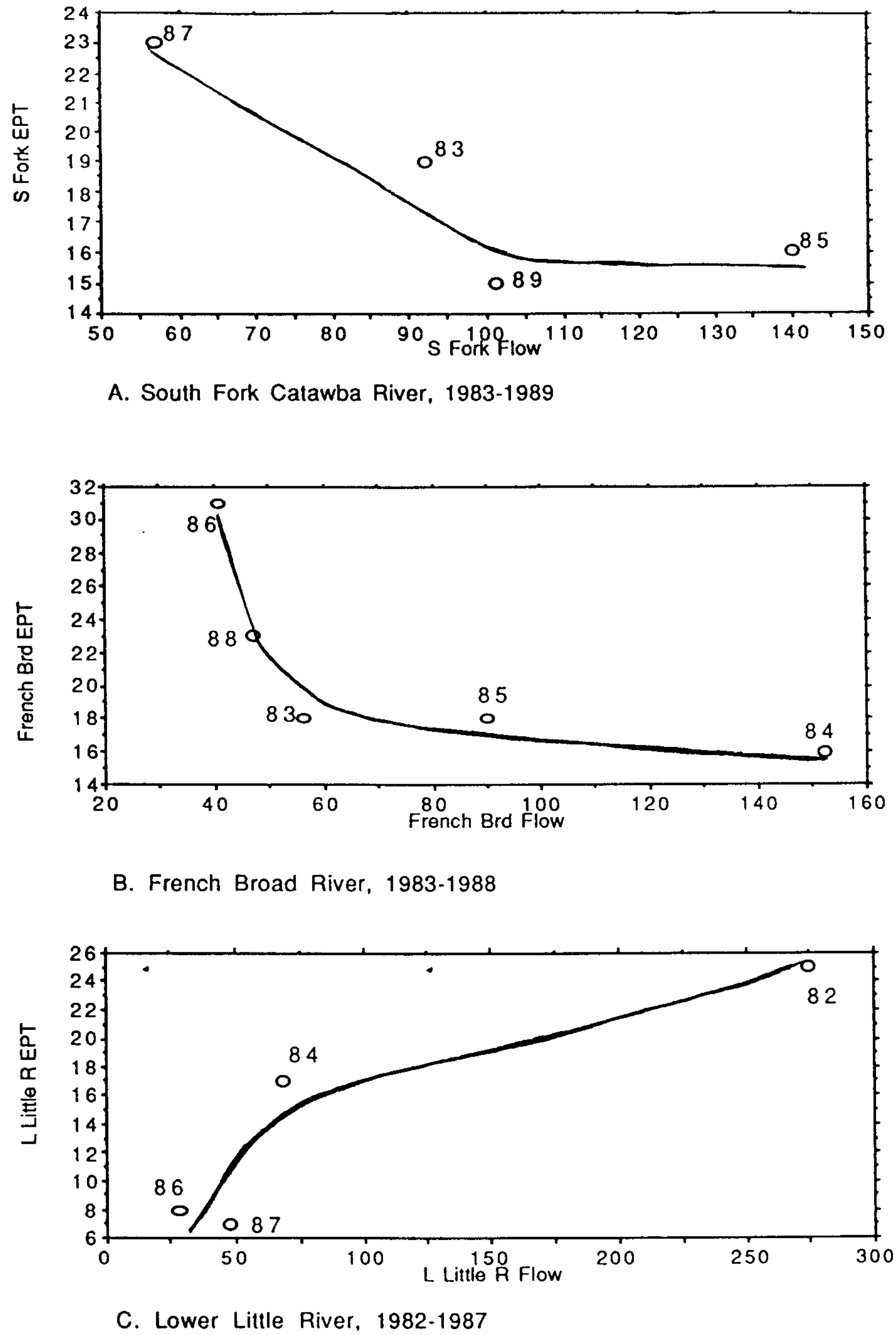

Figure 3. Examples of flow (as % of normal) vs. EPT Taxa Richness: South Fork Catawba River at MacAdenville, French Broad River at Marshall and Lower Little River at Manchester.

North Carolina has had such a network in place since 1983, and samples have been taken after both drought and flood conditions. A few examples have been drawn from this data base toillustrate possible complications caused by between-years changes in flow.

Figure 3 shows flow (as percent of average flow) for three sites. Two of these sites (Figure 3A and 3B) illustrate results from catchments affected by nonpoint runoff. For both the French Broad River at Marshall and the South Fork Catawba River at McAdenville, there was an inverse

relationship between flow and EPT taxa richness. Low flows, especially during the summers of 1987-1988, were associated with an increase in EPT taxa richness, but it is unlikely that this changes represent a true long-term change in water quality.
The third site is the Lower Little River at Manchester. There is a municipal wastewater treatment plant above this station, with a permitted flow of 8.0 MGD. During high flow years, (1982, 1984) relatively high EPT taxa richness values were recorded. Low flow years, however, provided little dilution for the wastewater discharge, and EPT taxa richness declined sharply. Changes in flow probably contribute to the decline in taxa richness at the Lower Little River site, although this information does not preclude the possibility of an actual decline in water quality as well.

Summary

Many factors affect the variability of benthic macroinvertebrate data. Much of this variability can be reduced by appropriate choices of sample sites, collection method, identification level, and analysis techniques. Variability can also be reduced by making corrections for predictable changes associated with habitat characteristics (ecoregion, stream size) or the time of the year. In the absence of specific corrections methods, analyses should be supported by a comparison with ecoregion reference sites. The effects of changes in flow are less predictable that habitat associated changes, but the general trend can be evaluated based on land use, ecoregion, stream size and the presence of point source dischargers.

North Carolina is in the process using a computerized data base to correct biocriteria for predictable variation in taxa richness based on ecoregion, stream size and seasonal changes. Collections in very small streams or during spring months can be expected require some adjustment before applying biocriteria. Our data base is also being used to derive tolerance values for a Hilsenhoff-type biotic index.

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